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Assessment of co-composting of sludge and woodchips in the perspective of environmental impacts (EASETECH)

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Abstract

To reveal potential impacts to environment and human health quantitatively, co-composting and utilization of sludge and woodchips were investigated using a life-cycle-based model, EASETECH. Three scenarios were assessed through experiments using different material ratios. Emission amounts during co-composting were determined by monitoring data and mass balance. With 100 t sludge treatment, co-composting showed impacts to acidification (29.9 PE) and terrestrial eutrophication (57.7 PE) mainly for ammonia emission. Compost utilization presented savings on freshwater eutrophication (–1.5 PE) because of phosphorus substitution. With the application of fewer woodchips, impacts to acidification and terrestrial eutrophication decreased because more ammonium was reserved rather than released. All impacts to human toxicity were not significant (8.2±0.6 PE) because the compost was used for urban landscaping rather than farming. Trace gaseous compounds showed marginal impacts to global warming and toxicity categories. The results provide a new perspective and offer evidence for appropriate sludge treatment selection.

Key words

Co-composting; sludge; environmental impact; life cycle assessment (LCA); EASETECH

1. Introduction

With the growth of urbanization, the generation of sewage sludge is increasing rapidly. In China, over 20 million tons of sewage sludge (wet weight) is produced per year (Xu et al., 2014). In Europe, the number is even larger as more than 10 million tons of sewage sludge in dry matter is produced every year (Rodriguez et al., 2012). Sludge disposal options are normally adjusted to local conditions, including geographical, legal, and economic circumstances, with the most widely available ones being agriculture utilization, waste disposal sites, land reclamation and restoration, incineration, and other novel uses (Fytili & Zabaniotou, 2008). Before utilization and disposal, municipal sludge normally has to be dewatered and/or treated to eliminate the bacteria, viruses, and organic pollutants; many technologies including dewatering, anaerobic digestion, and aerobic composting have thus been developed (Dong et al., 2014). Among these technologies, composting followed by land application is one of the most appropriate ways for economical sludge treatment and disposal (Wong et al., 2011).

However, because of its compacted structure, high water content, and low C/N ratio, municipal sludge can hardly be composted by itself (Banegas et al., 2007). Co-composting of municipal sludge and other materials, including municipal solid waste (Lu et al., 2009), saw dust (Yousefi et al., 2013), and food industry waste (Ammari et al., 2012), is therefore promising given their complementary characteristics. Garden waste, which normally has loose structure, low water content, and high C/N ratio, is widely applied in co-composting with sludge (Albrecht et al., 2010; El Fels et al., 2014). However, considering that one of the main aims of the bioprocess is to treat waste and reduce its environmental impacts, pollutant emissions and environmental impacts during the co-composting are always key concerns. Of utmost concern is the fact that land application of sewage sludge and its compost entails risks on ecological safety due to potential accumulation of toxic elements (Singh & Agrawal, 2008; Sreesai et al., 2013).

To better understand the potential impacts to the environment, life cycle assessment (LCA) of sewage sludge treatment has been gaining ground (Yoshida et al., 2013). LCA can systematically and effectively evaluate the potential environmental burden associated with energy consumption, process, product, and substitution during sludge treatment (Hong et al., 2013). In the current paper, based on pilot experiments focusing on parameters such as material ratios, temperatures, and changes in water content, the co-composting of municipal sludge and garden waste was assessed in the perspective of environmental impacts by using a newly developed LCA-based tool called EASETECH. This model software can perform life cycle assessment of complex systems involving different environmental technologies in the perspective of environmental impacts, with especial professional function for solid waste system modeling. By using this model, the current study can reveal the life cycle inventories and impact potentials of different co-composting operations, by investigating and assessing their emissions and material and energy consumption. The results provide important supplement to technical study for better understanding the environmental benefits or burdens of the co-composting process, as well as provide a new perspective and offer evidence for choosing the proper operations or technologies for municipal sludge treatment.

2. Materials and Methods

2.1. Composition of materials

The municipal sludge used in the current study was produced and dewatered in a municipal wastewater treatment plant in Suzhou, China. Garden waste was collected from urban landscaping projects, from which clipped branches were selected and crushed into woodchips that were 2–3 cm in length and 3 mm thick. The compositions of the sludge and woodchips were analyzed in the laboratory prior to composting, as shown in Table 1.

Table 1 Compositions of raw sludge and woodchip

Item	Sludge	Woodchip
Water (%)	84.63±0.01	24.73±0.12
VS (%TS)	62.24±0.27	94.05±1.33
C (%TS)	29.17±0.28	37.34±2.18
H (%TS)	4.78±0.07	5.02±0.23
N (%TS)	4.40±0.08	1.64±0.33
Cd (%TS)	1.717×10^{-3}	not detected
Cr (%TS)	5.620×10^{-3}	2.786×10^{-3}
Cu (%TS)	1.623×10^{-2}	2.647×10^{-3}
K (%TS)	0.6516	0.7348
Ni (%TS)	2.771×10^{-3}	2.169×10^{-3}
P (%TS)	1.260	5.721×10^{-2}
Pb (%TS)	1.473×10^{-3}	5.352×10^{-2}
Zn (%TS)	6.291×10^{-2}	9.353×10^{-3}

2.2. Co-composting technology description and experimental design

Pilot experiments of co-composting were carried out in a biotechnology company with a treatment capacity of 100 t sewage sludge per day in Suzhou, China. Windrow process was used with turning over by an upender. The sludge and woodchips were first weighed and mixed until well-distributed. Subsequently, over 5 tons of the mixed waste was windrowed, with length, width, and height of 5, 1.8, and 1 m. Co-composting processes were operated for 45 days, during which the mixed waste was turned over about once in every 4 days.

Three experimental batches (A, B, and C) of co-composting were implemented, with mass ratios (in wet weight) between sludge and woodchips of 3:1, 4:1, and 5:1, respectively. During the co-composting, the temperatures, percentages of CH_4 and CO_2, and releasing rates of NH_3 were measured once per day in the first 30 days, and then once every 2 days in the remaining 15 days given that the decomposition rates became slower in the second half of the periods. For the same reason, the Volatile solid (VS) of the mixed waste was measured once every 2 days in the first 30 days and once every 4 days in the remaining 15 days by mixing of triple parallel samples. The concentrations of CO_2 and CH_4 were monitored daily. The daily distribution of CO_2 and CH_4 (the percentage of everyday

CO_2 and CH_4 amounts in terms of total volume of CO_2 and CH_4) was thus calculated according to the VS monitoring data during the processes, with the idea that the volume of CO_2 and CH_4 was produced from VS decomposition proportionally. The water content, VS, and compositions, including nutrient elements, heavy metals, and germination indexes of each batch, were analyzed at the end of the experiments. Based on the element analysis before and after co-composting experiments, the proportions of C and N in the sludge, woodchips, and compost can be determined and used to calculate the C and N losses. Furthermore, gaseous emissions during the first batch of co-composting were parallel sampled using polyester bags and analyzed to reveal the impact contributions of the trace gases. Thirty odorous pollutants, such as toluene, dimethyl sulfide, limonene, and 1,2-dichloro-ethane, were determined as shown in Section 3.3.

2.3. Analysis method

The water contents and VS of the waste were determined by weight method using a drying oven and a muffle furnace. Elements of C, H, and N were analyzed using an Elemental Analyzer CE440 (Exeter Analytical, Inc., USA). Elements of P, K, and heavy metals listed in Table 1 were analyzed through inductively coupled plasma–atomic emission spectrometry (ICP-AES, IRIS intrepid, Thermo Electron Co., USA). Concentrations of CH_4 and CO_2 were monitored *in situ* (20 cm beneath the surface to avoid air interference) by using a biogas analyzer (Geotech Biogas 5000, Shanghai Zhonglin Co., China). The release rates of NH_3 were measured by using a static chamber technique and a multiple gas analyzer (Dräger X-am 7000, Drägerwerk AG & Co., Germany). The trace compounds in gaseous emissions were analyzed by a gas chromatography–mass spectrometer (GC–MS) system (Agilent 7890A-5975C, Agilent Technologies, Inc., USA). The germination test was carried out and the germination index was calculated according to the method reported by Roca-Pérez et al. (2009). The temperatures and pH were monitored routinely.

2.4. Model description and scenario setup

Based on the experimental data from co-composting of sludge and woodchips, the data related to life cycle assessment of the processes were investigated and then modeled with an LCA-based model called EASETECH. EASETECH is an LCA model for the assessment of environmental technologies newly developed at the Technical University of Denmark (Clavreul et al., 2014). EASETECH can perform life cycle assessment of complex systems handling heterogeneous material flows, with professional function for solid waste system modeling. With a focus on material flow modeling, resource use and recovery, as well as environmental emissions associated with environmental management systems can be modeled in a life cycle context. Related data are first input for all the process libraries including waste generation, collection and transportation, various treatment and disposal technologies, resources and recovery technologies, and related upstream and downstream processes. Subsequently, scenarios are created by connecting related processes from the libraries to represent systems to be modeled. The program of EASETECH then uses data contained in the scenario to compute results (Clavreul et al., 2014). The results can be provided in four levels, namely, life cycle inventory, characterization, normalization, and weighting, presenting impacts to 10 environmental categories, including global warming 100 years (GW100), terrestrial acidification (AC), freshwater eutrophication (FEP), terrestrial eutrophication (TEP), marine eutrophication (MEP), stratospheric ozone depletion 100 years (OD100), photochemical oxidant formation (POF), ecotoxicity (ET), human toxicity carcinogenic (HT c), and human toxicity non-carcinogenic (HT nc). The reference factors for normalization are incomplete in China, so we provided the results with person equivalent (PE) in terms of the methods recommended by International Reference Life Cycle Data System (ILCD, 2011). Therefore, as in terms of the Chinese situation, comparison across categories in the current paper was for reference only. The detailed sources of the LCA methods have

been reported by Yoshida et al. (2014).

To assess the co-composting system for sludge and woodchip treatment in the perspective of environmental impacts, three scenarios were set up according to the three operational batches during the pilot experiments. Different from the experimental batches, the scenarios which were used for modeling referred to abstract systems involving the co-composting processes as well as the upstream and downstream processes. For comparison purpose, the same amounts of waste (100 ton of sludge, 33.33 ton of woodchips) were assigned for all the scenarios. The waste was mixed according to the mass ratios shown in Table 2, and then it was co-composted for 45 days with solid and gas monitoring (no leachate was detected on the impermeable ground). To make the systems complete and comparable, downstream processes were also included. As the downstream process in the scenarios, the compost was used for urban landscaping projects to substitute fertilizers. To compare the results, using compost on agricultural soil was also modeled in terms of Scenario A. The redundant woodchips in Scenarios B and C were assumed to be landfilled in a municipal solid waste landfill according to the usual disposal of garden waste.

Table 2 Scenarios representing the three batches and their waste flow

Waste flow	Scenario A	Scenario B	Scenario C
Mass ratio of Sludge and woodchips	3:1	4:1	5:1
Sludge for co-composting	100.00 t	100.00 t	100.00 t
Woodchips for co-composting	33.33 t	25.00 t	20.00 t
Woodchips to landfill	0.00 t	8.33 t	13.33 t

3. Results and Discussion

3.1. Mass balance of C and N loss and emissions during co-composting

Material flow analysis is the first step before LCA of waste. Besides flows of substances, the flows of key elements, such as C and N, are important for understanding the amounts of corresponding

emissions. All the C loss during the co-composting can be attributed to CO_2 or CH_4 emission, given that the trace gases only took marginal proportion and could be ignored. All the N loss should be attributed to NH_3, N_2O, or N_2 released to air. Table 3 shows the C and N losses calculated from the changes in the amount before and after the experiments. The maximum potentials of released CO_2 and CH_4, as well as NH_3, were also calculated based on C and N balances.

Table 3 C and N balances before and after the co-composting experiments

Scenario	A	B	C
Mixture amount (t)	133.33	125.00	120.00
C in mixture (t)	9.90	10.75	8.18
N in mixture (t)	0.777	0.920	0.747
Compost amount (t)	29.12	36.85	34.40
C in compost (t)	3.18	3.10	2.93
N in compost (t)	0.372	0.402	0.382
C loss (t)	6.72	7.66	5.25
N loss (t)	0.405	0.518	0.365
Maximum potentials of CO_2 and CH_4 (m^3)	12540	14292	9795
Maximum potential of NH_3 (m^3)	648	829	585

Based on the maximum potentials, the daily distribution, and the concentrations of CO_2 and CH_4, the daily amounts of CO_2 and CH_4 were calculated. Given the large amount of data, the daily emissions of CO_2 and CH_4 were not shown. Table 4 shows the summations of daily CO_2 and CH_4 production during co-composting processes for each scenario. The release rates of NH_3 were used to calculate the daily flux of NH_3 during co-composting. Similarly, the summation of daily NH_3 production for each scenario is also listed in Table 4.

Table 4 The emissions of CO_2 and CH_4 during the co-composting processes

Scenario	Total amount of CO_2 (m^3)	Total amount of CH_4 (m^3)	Total amount of NH_3 (m^3)
A	12540	0	647
B	14067	225	393
C	7737	2058	8

The total amount of NH_3 from co-composting process in Scenario A was almost equal to the calculated maximum potential of NH_3 shown in Table 3, indicating the N balance and the very low

percentages of N_2O and N_2 production. However, those from Scenarios B and C presented significant differences compared to the calculated maximum potentials of NH_3. This finding was probably due to the generated ammonia that did not evaporate efficiently, and thus, remained in the water of the waste as NH_4^+. This supposition was proven by the temperature monitoring during the co-composting. According to the experimental data of Batch A, the temperature during co-composting increased rapidly to 66 °C at the beginning, and the temperatures above 55 °C lasted for 21 days, indicating efficient decomposition of organic waste. Batch B also presented an effective co-composting, but the temperatures above 55 °C lasted for only 12 days. However, due to the high content of municipal sludge in Batch C and the low ambient temperature, the activity of aerobic microorganisms did not behave normally and the temperatures varied in the range of 15 °C to 30 °C. Therefore, the water evaporation was restricted and the generated ammonia can hardly be released. It was thus deduced that the N loss in the total solid was mainly attributed to ammonia dissolved in the water, which was not included in the air emissions.

Based on the monitoring data and mass balance of C and N loss, major emissions including CO_2, CH_4, NH_3, N_2, and NH_4^+ can be determined for each co-composting batches, which were then input into EASETECH for scenario modeling.

3.2. Environmental impacts from co-composting of sludge and woodchips

3.2.1. Environmental impacts from Scenario A

According to the experimental phenomena, Batch A achieved a successful co-composting process. Accordingly in Scenario A, after the co-composting of 133.3 t of sludge and woodchips, 29.1 t of compost can be produced, with most of the moisture removed. From the perspective of environmental impact, the whole process has potential impacts to HT c, HT nc, ET, TEP, AC, and so on, as shown in

Figure 1. According to the life cycle inventory, almost all of the heavy metals, such as Zn (11.8 kg), Cu (3.1 kg), and Ni (0.9 kg), from sludge and woodchips will remain in the soil due to compost utilization. The concentrations of these elements are in the same order of magnitude to those in compost from sewage sludge and grass clippings reported by Sreesai et al. (2014). These heavy metals have significant potential of impacting ET (527.2 PE) and HT nc (8.4 PE) and HT c (0.4 PE). This is in accordance with the finding from sludge LCA reported by Xu et al. (2014). However, the heavy metal speciation, which may affect their toxicity significantly, was not discriminative during LCA modeling. Further germination test carried out in our lab indicated that, the germination index increased to 70.9% with the compost compared to 40.7% before co-composting, indicating that co-composting is capable of reducing the toxicity in sludge.

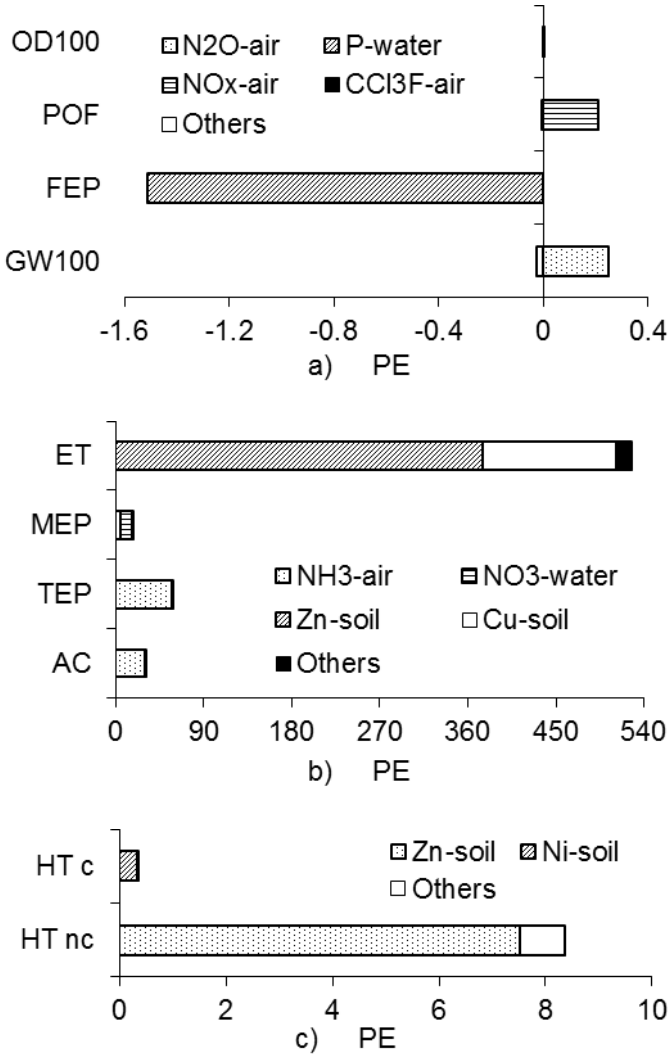

Figure 1. Normalized impact potentials of Scenario A in terms of substance style (a, b, and c are used to illustrate different coordinate ranges)
GW100: global warming 100 years; AC: terrestrial acidification; POF: photochemical oxidant formation; OD100: stratospheric ozone depletion 100 years; FEP: freshwater eutrophication; TEP: terrestrial eutrophication; MEP: marine eutrophication; ET: ecotoxicity; HT c: human toxicity carcinogenic; HT nc: human toxicity non-carcinogenic.

A total of 491 kg of ammonia (647 m^3) released into the air during the co-composting is the major contributor to AC (29.9 PE) and TEP (57.7 PE). By contrast, although 12540 m^3 of CO_2 is released during the process, the co-composting together with the compost utilization shows marginal impacts to GW100 because the CO_2 emission is totally attributed to organic decomposition, in which carbon is ultimately from photosynthesis with CO_2 in air. Thus, it is considered neutral to GW100. A small

amount of nitrogen is lost as N_2O, resulting in impacts to GW100 with 0.3 PE. In particular, the compost utilization presents obvious advantages in saving FEP (–1.5 PE) because of substitution of P fertilizer. Conversely, substitution of N fertilizer does not benefit FEP improvement because N is not the limiting factor of eutrophication in freshwater, but is considered to result in potential MEP due to the effects of ammonia (4.8 PE) and nitrate (11.9 PE). Nevertheless, MEP is not applicable to the local situation which does not involve an ocean.

Focusing on the impacts from the co-composting process, process emissions only dominate impacts to AC (29.9 PE), TEP (57.7 PE), and MEP (4.8 PE). The impacts to the other categories are mainly attributed to fuel consumption of heavy equipment, such as the upender and dozer. According to the operation data, 5 L of diesel will be consumed to treat 1 t of mixed waste. Therefore, the normalized impact to GW100 is 0.25 PE from fuel consumption, which is four times higher than that from process emission. The impacts to POF (0.27 PE), OD100 (4.57×10^{-5} PE), ET (0.05 PE), HT c (4.62×10^{-4} PE), and HT nc (2.52×10^{-3} PE) are totally from fuel consumption. However, compared to those from compost utilization, the impacts from fuel consumption are insignificant, indicating that the critical points for pollution control in the whole process are use-on-land for ET and HT nc and HT c, and process emission for AC and TEP.

3.2.2. Environmental impact comparison among Scenarios A, B, and C

Different co-composting batches presented different reaction processes, compost characteristics, and emissions. Similar to Batch A, Batch B showed successful VS degradation and temperature rise. However, Batch C did not achieve the goal of composting according to its temperature rise, moisture evaporation, VS degradation, and methane production. Instead, anaerobic reaction was observed in Batch C. Accordingly, the released methane in Scenario C was increased and was one of the major contributors to GW100, as shown in Table 5.

Table 5 The comparison of normalized impact potentials from the three scenarios

Category	Scenario A	Scenario B	Scenario C
GW100	0.2	2.7	8.1
AC	30.0	18.3	0.5
POF	0.2	0.4	0.7
OD100	4.6×10^{-5}	4.4×10^{-5}	4.2×10^{-5}
FEP	-1.5	-1.5	-1.5
TEP	58.2	35.5	1.2
MEP	17.2	45.0	51.3
ET	527.2	498.7	481.5
HT c	0.4	0.3	0.3
HT nc	8.4	7.8	7.4

The life cycle inventory of the three scenarios further indicates that the amounts of methane from organics are 0, 841.1, and 2559.0 kg, respectively. However, co-composting processes only account for part of them due to the anaerobic reaction. The other parts in Scenarios B and C (1.9 PE and 3.0 PE, respectively) are mainly attributed to the real anaerobic degradation of redundant woodchips in landfill, as shown in Figure 2. Garden waste is normally landfilled if it is not used as auxiliary materials like woodchips in co-composting. The emissions such, as methane, will impact both GW100 and POF. Therefore, the avoided emissions from landfilling garden waste should be considered one of the advantages of the co-composting process.

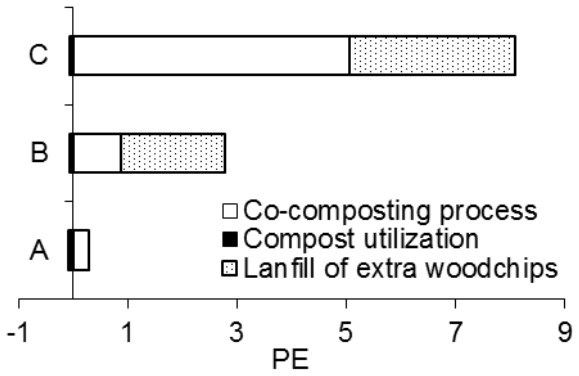

Figure 2. Normalized impact potential to global warming (GW100) of processes in the three scenarios

Part of the ammonia produced during processing in Scenarios B and C remains as ammonium in liquid phase, which mainly impacts MEP (32.2 and 39.1 PE, respectively), as shown in Figure 3. The impacts from compost utilization do not show significant difference among the three scenarios

because the nitrogen amounts left in the compost are not much different from each other (see Table 3). The distribution of nitrogen during utilization is also similar. According to the life cycle inventory and normalized results in substance style, the nitrate run off into surface and ground water is the main contributor to MEP during utilization (12.0±0.6 PE in all the scenarios).

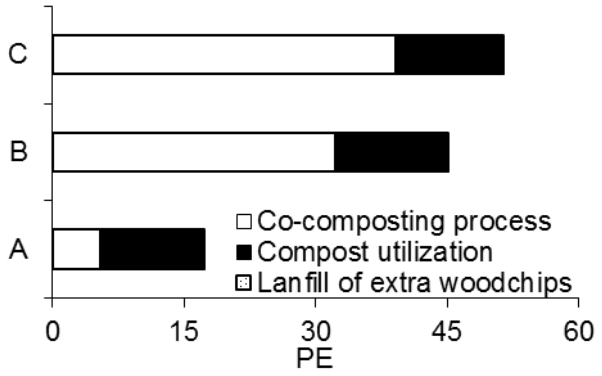

Figure 3. Normalized impact potential to marine eutrophication (MEP) of processes in the three scenarios

Meanwhile, the impacts to AC and TEP in Scenario C are much lower than those in Scenarios A and B (Table 5), which benefit from the less ammonia released into the air. Moreover, ET, HT c and HT nc slightly decreased from 527.2, 0.4, and 8.4 PE in Scenario A to 481.5, 0.3, and 7.4 PE in Scenario C. The decrement is attributed to the fact that less heavy metal will remain in the soil when fewer woodchips are applied. Thus, less compost is used on the land. This phenomenon is ascribed to the fact that some of the heavy metals in woodchips are comparable to those in sludge, according to the data in Table 1. The pollutants in the redundant woodchips in landfill are considered relatively safe to the environment and to humans, although they still pose risks of release into the environment someday. In the LCA of sludge management, compost with high concentrations of heavy metals is normally reported to exhibit high impacts to the soil as HT c, HT nc and ET (Dong et al., 2014). However, the impacts to human toxicity and the difference among scenarios are not that significant in the current study because the compost is supposed to be used for urban landscaping projects rather than farming purpose. Table 6 shows the possible results for human toxicity if the compost is used for agricultural

soil in Scenario A. Compared to forestry soil, the characterization factors for heavy metals in agricultural soil, which are also listed in Table 6, are normally much higher due to their potential of entering the food chain. This result indicates that the utilization ways of compost derived from sludge should be paid special attention. Using it for landscaping is less risky and more acceptable in the perspective of environmental impact.

Table 6 Comparison of heavy metal behaviours when using compost on farmland and landscaping in Scenario A

Category	Heavy metal	Characterization factor for agricultural soil	Characterization factor for forestry soil	Normalized potential with using on farmland	Normalized potential with using on landscaping
HT c	Cd	4.80×10^{-4}	8.08×10^{-7}	2.10	3.54×10^{-3}
	Cr	-	-	0	0
	Cu	-	-	0	0
	Ni	1.10×10^{-4}	1.97×10^{-5}	1.92	0.34
	Pb	7.60×10^{-5}	2.02×10^{-7}	19.13	0.05
	Zn	-	-	0	0
HT nc	Cd	0.13	2.17×10^{-4}	28.08	0.05
	Cr	1.80×10^{-9}	1.52×10^{-9}	4.47×10^{-7}	3.78×10^{-7}
	Cu	3.74×10^{-5}	4.55×10^{-7}	0.11	1.29×10^{-3}
	Ni	5.90×10^{-6}	1.11×10^{-6}	5.08×10^{-3}	9.55×10^{-4}
	Pb	0.027	7.08×10^{-5}	334.90	0.88
	Zn	0.044	7.02×10^{-4}	472.10	7.53

3.3. Environmental impacts caused by the trace compounds in air emissions

Considering that an LCA based on routine data collection may underestimate impact potentials due to limitations of substance coverage (Yoshida et al., 2014), trace compounds in air emissions during the co-composting were sampled and analyzed through GC–MS with the ability of identifying over 120 kinds of possible trace compounds. In Scenario A, 30 trace compounds were identified, among which 24 substances are included in the environmental impact categories used in this study, as shown in Figure 4. The rest six compounds included dimethyl sulfide, α-pinene, β-Pinene, limonene, isobutane, and 2-methyl-butane. The total amount of each trace compound was calculated based on measured

concentration and daily gas volume, calculated by using the daily amount and proportion of CO_2. The conventional air emissions during co-composting only show impact potentials to GW100, AC, and TEP. However, these trace compounds contribute to the categories of GW100, POF, OD100, ET, HT c and HT nc. In these categories, dichlorodifluoro-methane (1.5×10^{-3} PE), propene (5.5×10^{-5} PE), trichlorofluoro-methane (1.6×10^{-2} PE), carbon disulfide (2.4×10^{-6} PE), tetrachloro-methane (7.5×10^{-5} PE), and carbon disulfide (3.0×10^{-5} PE) are the dominant contributors, respectively. The normalized impact potentials from most of the trace compounds are in the order of 10^{-8} to 10^{-4}, as shown in Figure 4. For example, the impact to GW100 from the trace pollutants is 1.9×10^{-3} PE, which is insignificant compared to that from co-composting emissions in Scenario A (0.05 PE). As to the categories of POF, ET, and HT nc, the impact potentials from fuel consumption during co-composting (10^{-3} to 10^{-1} PE) are far higher than those from the trace pollutants (10^{-6} to 10^{-4} PE). The impact potential to OD100 is the only exception, which is 1.8×10^{-2} PE from the trace compounds and much higher than that from fuel consumption (4.6×10^{-5} PE). From this point of view, these trace gases are not very important in the whole process of sludge and garden waste co-composting.

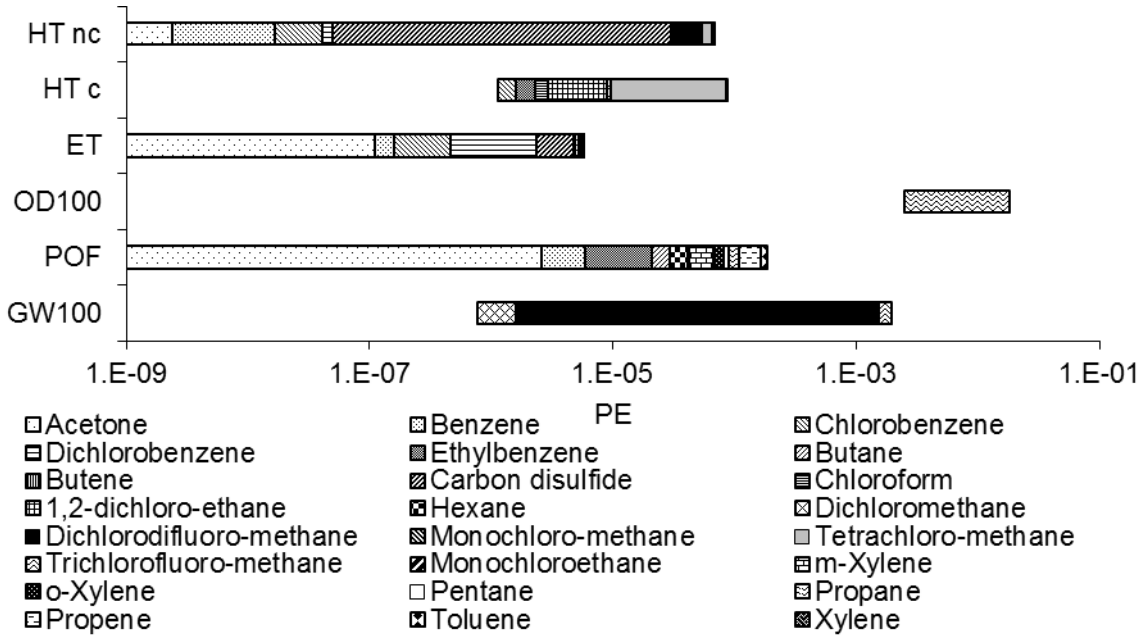

Figure 4. Normalized impact potentials from the trace gaseous pollutants during co-composting in Scenario A

GW100: global warming 100 years; POF: photochemical oxidant formation; OD100: stratospheric

ozone depletion 100 years; ET: ecotoxicity; HT c: human toxicity carcinogenic; HT nc: human toxicity non-carcinogenic.

However, according to our study on odorant pollution from waste treatment facilities, all of the abovementioned trace compounds are odorant sources, resulting in an unpleasant feeling in the surrounding population. These pollutants have different odor thresholds, and thus, perform differently in causing odorant pollution. For example, carbon disulfide and n-pentane possess odor thresholds of 0.21 ppm and 1.4 ppm according to Nagata (2003). This suggested that carbon disulphide would contribute to odor pollution approximately 6 times higher than n-pentane because of its lower threshold, although they were measured as similar concentrations and releasing amounts. This kind of environmental impact is in the local scale and is normally not included in LCA. The relevant study on odor pollution evaluation has been published in another article (Zhao et al., 2014), and the corresponding function embedding in LCA-based model is being undertaken by the authors.

4. Conclusions

Three scenarios based on co-composting experiments of sludge and woodchips were investigated using the LCA-based model, EASETECH, to reveal the environmental impacts. Co-composting of 100 t sludge and 33.33 t woodchips impacts AC (29.9 PE) and TEP (57.7 PE) by ammonia emission. Compost utilization benefits FEP (–1.5 PE) by phosphorus substitution. Fewer woodchips lead to lower impacts to AC and TEP because more ammonia is reserved as ammonium. The impacts to HT c and HT nc are not significant (8.2±0.6 PE) when applying the compost to landscaping. The results provide new perspective and offer evidence for appropriate selection of sludge treatment options.

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Figure Captions

Figure 1. Normalized impact potentials of Scenario A in terms of substance style (a, b, and c are used to illustrate different coordinate ranges)

GW100: global warming 100 years; AC: terrestrial acidification; POF: photochemical oxidant formation; OD100: stratospheric ozone depletion 100 years; FEP: freshwater eutrophication; TEP: terrestrial eutrophication; MEP: marine eutrophication; ET: ecotoxicity; HT c: human toxicity carcinogenic; HT nc: human toxicity non-carcinogenic.

Figure 2. Normalized impact potential to global warming (GW100) of processes in the three scenarios

Figure 3. Normalized impact potential to marine eutrophication (MEP) of processes in the three scenarios

Figure 4. Normalized impact potentials from the trace gaseous pollutants during co-composting in Scenario A

GW100: global warming 100 years; POF: photochemical oxidant formation; OD100: stratospheric ozone depletion 100 years; ET: ecotoxicity; HT c: human toxicity carcinogenic; HT nc: human toxicity non-carcinogenic.